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Water Quality Analysis of a Freshwater Diversion at Caernarvon, Louisiana

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ABSTRACT: Since 1991, Mississippi River water has been diverted at Caernarvon, Louisiana, into Breton Sound estuary. Breton Sound estuary encompasses 1100 km^2 of fresh and brackish, rapidly subsiding wetlands. Nitrite + nitrate, total Kjeldahl nitrogen, ammonium, total phosphorus, total suspended sediments, and salinity concentrations were monitored at seven locations in Breton Sound from 1988 to 1994. Statistical analysis of the data indicated decreased total Kjeldahl nitrogen with associated decrease in total nitrogen, and decreased salinity concentrations in the estuary due to the diversion. Spring and summer water quality transects indicated rapid reduction of nitrite + nitrate and total suspended sediment concentration as diverted Mississippi River water entered the estuary, suggesting near complete assimilation of these constituents by the ecosystem. Loading rates of nitrite + nitrate (5.6–13.4 g m^{-2} yr^{-1}), total nitrogen (8.9–23.4 g m^{-2} yr^{-1}), and total phosphorus (0.9–2.0 g m^{-2} yr^{-1}) were calculated along with removal efficiencies for these constituents (nitrite + nitrate 88–97%; total nitrogen 32–57%; total phosphorus 0–46%). The low impact of the diversion on water quality in the Breton Sound estuary, along with assimilation of TSS over a very short distance, suggests that more water may be introduced into the estuary without detrimental affects. This would be necessary if freshwater diversions are to be used to distribute nutrients and sediments into the lower reaches of the estuary, in an effort to compensate for relative sea-level rise, and reverse the current trend of rapid loss of wetlands in coastal Louisiana.

Introduction

Eustatic sea-level stabilized near its present level after the last glaciation between 5,000 to 7,000 years ago (Milliman and Emory 1968). Since that time, sedimentation from the Mississippi River created a series of overlapping deltaic lobes that presently form the Mississippi deltaic plain in coastal Louisiana (Scruton 1960; Roberts 1997). Each deltaic lobe went through a cycle of active progradation followed by river abandonment and subsequent deterioration and submergence due to the combined effects of shore-face erosion and coastal subsidence (Penland et al. 1990). During the abandonment phase, sediment and nutrients were delivered to lobes adjacent to the river during spring floods (Hatton et al. 1983). These floods provided a source of mineral sediments, which contributed directly to vertical accretion; nutrients associated with these sediments promoted further vertical accretion through organic soil formation from wetland plant production (DeLaune et al. 1983). These increases in vertical accretion helped maintain marsh elevation above relative sea-level rise (RSLR), the combined effect of eustatic sea-level rise (1–2 mm yr^{-1}, Gornitz et al. 1982) and coastal subsidence. RSLR in the Mississippi delta is in excess of 10 mm yr^{-1} (Penland and Ramsey 1990). Other deltas of the world, such as the Nile and Rhone deltas are also experiencing high relative sea level (Stanley 1988; L'Homer 1992).

Since the early 1900's most wetlands in the Mississippi delta have been hydrologically isolated from the river by the construction of flood control levees (Mossa 1996). These levees prevent seasonal flooding and thus introduction of sediments and nutrients into nearby wetlands. Resuspended sediment from bay bottoms and reworked deltaic deposits have replaced fluvial sources of inorganic sediments in many regions of coastal Louisiana (Hatton et al. 1983). As a restoration effort, the State of Louisiana has developed a plan of freshwater diversions that will mimic flooding events of

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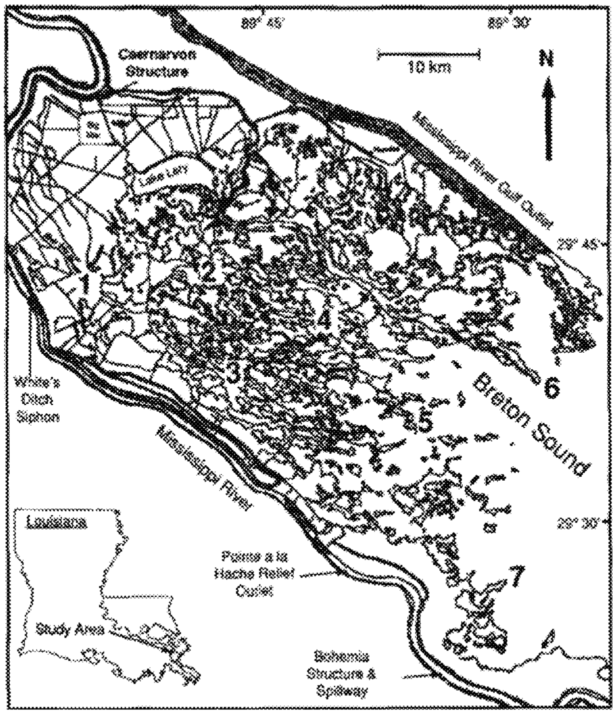

Fig. 1. Breton Sound Estuary. Numbers refer to water quality monitoring stations used for this analysis. The Caernarvon, White's Ditch Siphon and Bohemia structures are controlled freshwater diversions, and the Pointe a la Hache relief outlet and Bohemia spillway are areas of seasonal flooding of the Mississippi River.

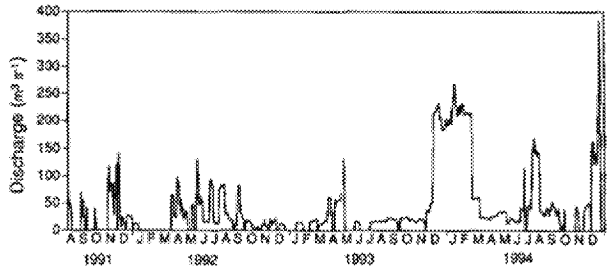

Fig. 2. Caernarvon diversion discharge schedule from August 1991 through December 1994.

the Mississippi River (Chatry and Chew 1985). When freshwater diversions were planned over two decades ago, the primary goal was to reduce salinity to enhance oyster production in surrounding regions (Chatry et al. 1983). More recently, diversions have increasingly been used as a way of delivering sediments and nutrients to wetlands in an attempt to counter RSLR (Day and Templet 1989).

There has been controversy about the effects of diverting Mississippi River water into coastal wetlands because of possible eutrophication as currently observed in Louisiana's offshore waters (Turner and Rabalais 1991; Rabalais et al. 1994). Eutrophication in the estuary itself is less likely because environments with high turbidity caused by river inputs of suspended sediments usually have low phytoplankton productivity due to low light availability (Cloern 1987). Wetlands are natural sinks for nutrients (Hatton et al. 1982; Sharp et al. 1982; Reddy et al. 1993), and thus represent a viable mechanism for decreasing the nutrient load of river water prior to reaching offshore. There are a number of changes which take place in the chemistry of river water as it flows into an estuary and mixes with sea water. Nitrate-nitrogen, for example, can be lost to the atmosphere through denitrification, assimilated into organic matter, or reduced to ammonium, with all of these processes occurring simultaneously.

Six freshwater diversions are currently in operation in Louisiana and several others are in various stages of evaluation or construction. The subject of this study is the diversion at Caernarvon, Louisiana, located approximately 20 km down river from New Orleans. The objective of this study is to analyze the nutrient and sediment dynamics that occurred in a Louisiana estuary in response to the addition of Mississippi River water. We hypothesized that there would be rapid assimilation of suspended sediments and nitrate, with associated increases in ammonium and total Kjeldahl nitrogen, and low uptake of phosphorus in the estuarine waters of Breton Sound.

Study Site

The Caernarvon freshwater diversion is the largest of six diversions currently in operation on the lower Mississippi River south of New Orleans. The diversion structure is located on the east bank of the Mississippi River near Caernarvon, Louisiana, at river mile 81.5 (Fig. 1). The water control structure is a five box culvert with vertical lift gates; each culvert is 4.6 m wide. The structure has the capability of passing 226 $m^3 s^{-1}$ of water. The Caernarvon control structure was completed in 1991 and freshwater discharge began in August of that year. Freshwater discharge from August 1991 to December 1993 averaged 21 $m^3 s^{-1}$ (Fig. 2). An average of 212 $m^3 s^{-1}$ flowed into the estuary from December 1993 through February 1994. The rest of 1994 had an average flow of 45 $m^3 s^{-1}$.

The Caernarvon diversion delivers water into Breton Sound estuary, which consists of 1,100 km^2 of fresh, brackish, and saline wetlands (Fig. 1). The estuary is hydrologically bounded to the west by the Mississippi River levee, to the north by natural levees of Bayou La Loutre, and to the east by spoil banks of the Mississippi River Gulf Outlet. The estuary has open connection to the Gulf of Mexico. Breton Sound wetlands were formed several thousand years ago as part of the Plaquemines-St. Bernard delta complex (Scruton 1960; Roberts 1997). Since then, approximately half of the original wetlands have submerged due to shore-face erosion and coastal subsidence (Penland et al. 1988).

Prior to the construction of the Mississippi River levee system there have been several documented natural crevasses in the region surrounding Caernarvon, delivering vast amounts of river water into Breton Sound estuary (Russell 1936; Davis 1993). These natural crevasses most likely have occurred intermittently over a much longer period than the historical record documents. Due to an extremely high river stage height during the great flood of 1927, the Mississippi River levee at Caernarvon was intentionally destroyed in April of that year to relieve New Orleans from possible flooding (Barry 1997). The resulting artificial crevasse was 979 meters wide and diverted up to 9,200 $m^3 s^{-1}$ of water, equal to half of the mean flow of the Mississippi River, from the river into Breton Sound estuary for four months (Davis 1993; Barry 1997). The historical record, therefore, indicates that the Breton Sound estuary has experienced massive periodic inputs of Mississippi River water as part of its evolution to its current ecological state.

In addition to the Caernarvon diversion, there are several other contemporary sources of Mississippi River water to Breton Sound estuary. The White's Ditch Siphon is a freshwater diversion that delivers water to the northwestern portion of Breton Sound estuary (Fig. 1), and has a maximum flow capacity of 10 $m^3 s^{-1}$, but generally flows at a much lower rate. We assumed the White's Ditch Siphon did not greatly affect this study due to its low flow. The Bohemia structure is another freshwater diversion that has the same flow capacity as the Caernarvon structure, and delivers water into the southeastern portion of the study area. Mississippi River water also enters the southern portion of the study area at the Pointe a la Hache Relief Outlet and the Bohemia Spillway, but only during high river stage. The Bohemia structure and spillway and the Pointe a la Hache Relief Outlet greatly affects water quality in southern Breton Sound, as indicated by the data presented below, but the exact volume of water introduced by these waterways is not monitored and is unknown.

Materials and Methods

In this study, a long term data set of water quality parameters from 1988 to 1994 was analyzed along with data collected during two field trips in 1996. A three year pre-diversion survey was carried out from January 1988 to July 1991 to establish baseline conditions of the study area in Breton Sound estuary. Post-diversion monitoring continued from August 1991 to December 1994. Water quality was monitored monthly at seven locations (Fig. 1). Station 1 was located in Shell Bayou; station 2 at the west shore of Lake Petit; station 3 in River Aux Chenes at the pipeline canal north of First Bay; station 4 in Lake Cuatro Cabello at Pato Cabello Pass; station 5 in Bay Gardene at Bayou Lost; station 6 at the navigation light near Mozambique Point; and station 7 at the oil well platform northeast of California Point (COE 1995).

Water quality parameters measured were nitrite + nitrate (NO_2 + NO_3), ammonium (NH_4), total Kjeldahl nitrogen (TKN), total phosphorus (TP), total suspended sediments (TSS), dissolved oxygen (DO), and salinity. The Corps of Engineers (COE) was responsible for collecting and analyzing samples from stations 1, 3, 4, 6 and 7. The Louisiana Department of Environmental Quality (LDEQ) was responsible for collecting and analyzing water quality samples from stations 2 and 5. LDEQ did not measure NH_4 concentrations. TP measurements made by LDEQ were not included in this analysis due to inconsistencies in the data. Total phosphorus data taken by the LDEQ had consistently lower values than those reported by the COE (Lane 1998).

Surface water samples were collected from 10–20 cm depth in acid washed glass or plastic containers with Teflon coated lids (Greenburg et al. 1985). Water samples were stored at 4°C for preservation and transported to either COE or LDEQ analytical laboratories where both filtered and unfiltered samples were frozen until analysis. The samples were analyzed for nitrite + nitrate (NO_2 + NO_3), ammonium ($NH_4 - N$), total Kjeldahl nitrogen (TKN), Total Phosphorus (TP), total suspended sediments (TSS), and salinity (Greenburg et al. 1985). Samples analyzed for NO_2 + NO_3 and NH_4 were filtered in the laboratory using 0.45 um millipore filters and concentrations determined by automated colorimetric cadmium reduction and automated colorimetric phenate methods, respectively. TKN and TP were analyzed using acid-digestion methods described by Greenburg et al. (1985). Samples analyzed for TSS were filtered in the laboratory through pre-dried and weighed 0.45 um millipore filters. Filters were then dried for 24 h at approximately 15°C after filtering and weighed to determine the amount of suspended materials in the water.

Total nitrogen (TN) was calculated by adding NO_2 + NO_3 and TKN values. Monthly data was separated into the following: December through February was designated as winter; March through May as spring; June through August as summer; and September through November as Autumn.

A Before-After-Control-Impact (BACI) statistical design (Underwood 1994) was used to determine whether the project had an impact on each of the measured variables at the water quality monitoring stations in Breton Sound estuary. The Underwood model was modified to account for variation due to months and years; the years were treated as random effects and months as fixed effects. The time within Before-After effect was partitioned into a year within Pre/Post (fixed) effect and a month within year and Pre/Post (random) effect. A significant Pre/Post by Station interaction ($\alpha < 0.05$) was interpreted as an impact of the project on the measured variable. If an impact was found, station by station contrasts were performed to determine which stations were affected.

Preliminary analysis of the long term data set indicated that $NO_2 + NO_3$ and TSS decreased to background levels by the time the first water quality monitoring station was reached. Field data transects were performed to obtain finer spatial resolution in the extreme upper estuary, north of the first two stations. Discrete water samples were taken April 30 and June 21, 1996, at the diversion structure and along the two major bayous to stations 1 and 2. The same methods cited above were used to analyze the samples for $NO_2 + NO_3$ and TSS at the Coastal Ecology Institute at Louisiana State University.

Nutrient loading rate (expressed as g m^{-2} yr^{-1}) and removal efficiency (the percentage of nutrients removed from the overlying water column and retained within the wetland ecosystem or released into the atmosphere) were calculated for the portion of wetlands between the Caernarvon structure and the first two water quality stations. Data was used from 1992 through 1994, with each year analyzed separately. The area of wetlands was conservatively estimated to be 260 km^2. Approximately 33% of water discharged from the Caernarvon diversion flows west of Big Mar, eventually reaching station 1, the remaining water flows east into Lake Leary and eventually to station 2. This flow estimation was used in normalizing removal efficiencies, which were calculated by comparing nutrient concentrations in Mississippi River and stations 1 and 2.

Results

The BACI analysis, along with station by station contrasts, indicated lower salinity concentrations at all stations (Table 1), except those in the lower estuary (stations 5–7). Salinity concentrations ranged from 0 at the Mississippi River and steadily increased to means of 12–17‰ at the most distant stations (Fig. 3).

There was no significant impact of the diversion at all of the water quality monitoring stations for

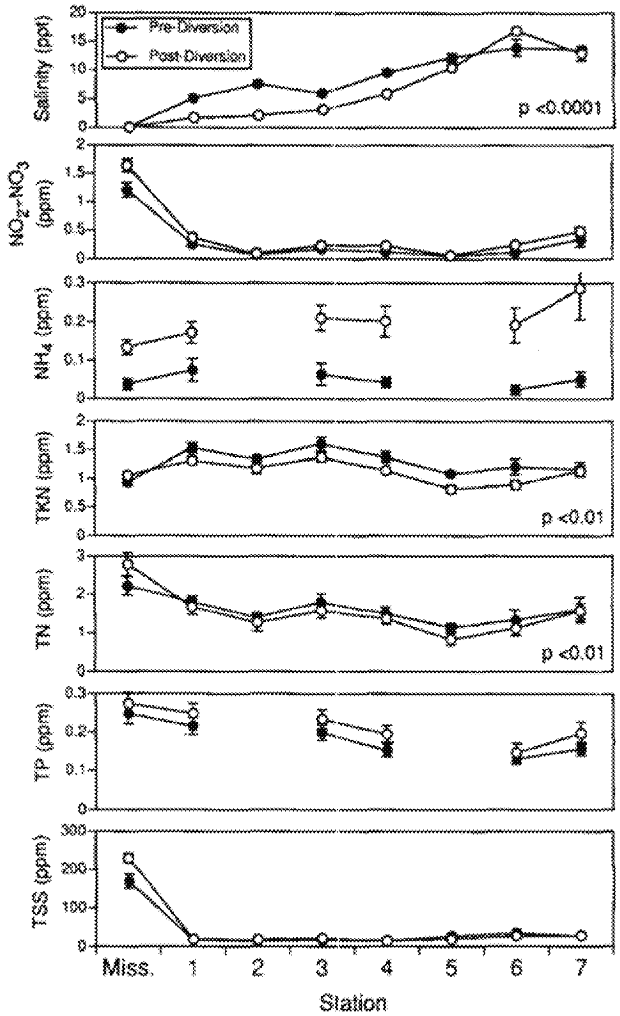

Fig. 3. Pre- and post-diversion water-quality station annual means with standard error bars, for salinity, nitrite + nitrate, ammonium, total Kjeldahl nitrogen, total nitrogen, total phosphorous, and total suspended solids. Significant results from BACI analysis are indicated.

$NO_2 + NO_3$ based on the results of the BACI analysis (Fig. 1). Station 7 had elevated $NO_2 + NO_3$ concentrations compared to the other marsh stations presumably due to the addition of river water in the region by the Bohemia structure and spillway (Fig. 3). Mean pre- and post-diversion Mississippi River water $NO_2 + NO_3$ concentrations ranged from 1.2 to 1.6 ppm, while the marsh water quality monitoring stations ranged from 0.1 to 0.5 ppm, suggesting rapid reduction in $NO_2 + NO_3$ levels as river water entered the estuary. This is supported by the salinity mixing diagram (Fig. 4), which indicated that the Breton Sound wetlands and shallow waters were acting as a strong sink for $NO_2 + NO_3$ (see Liss [1976] for mixing diagram discussion). When the post-diversion data was analyzed by seasons (Fig. 5), nitrite + nitrate concen-

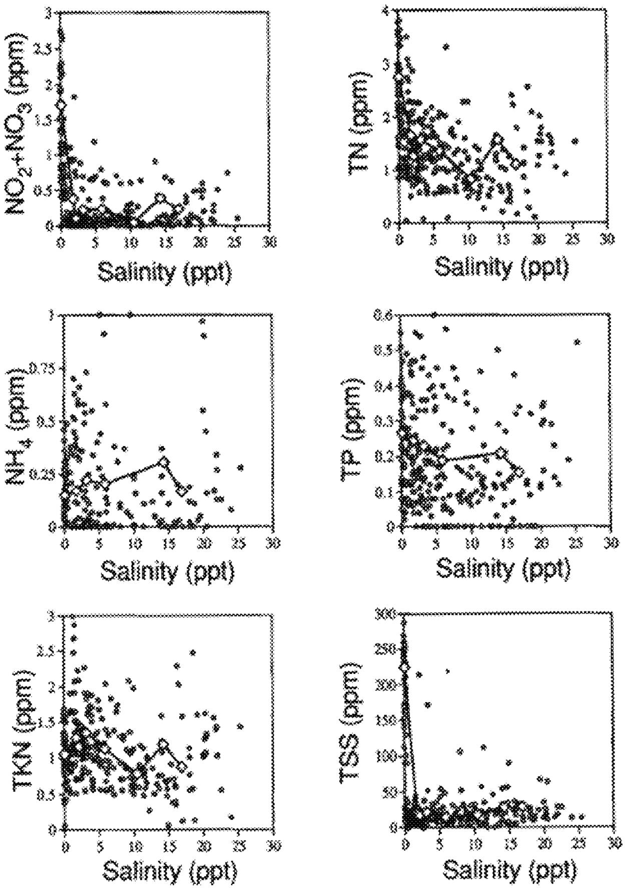

Fig. 4. Post-diversion salinity mixing curve with raw data points shown as circles and diamonds indicating overall means at each water quality station.

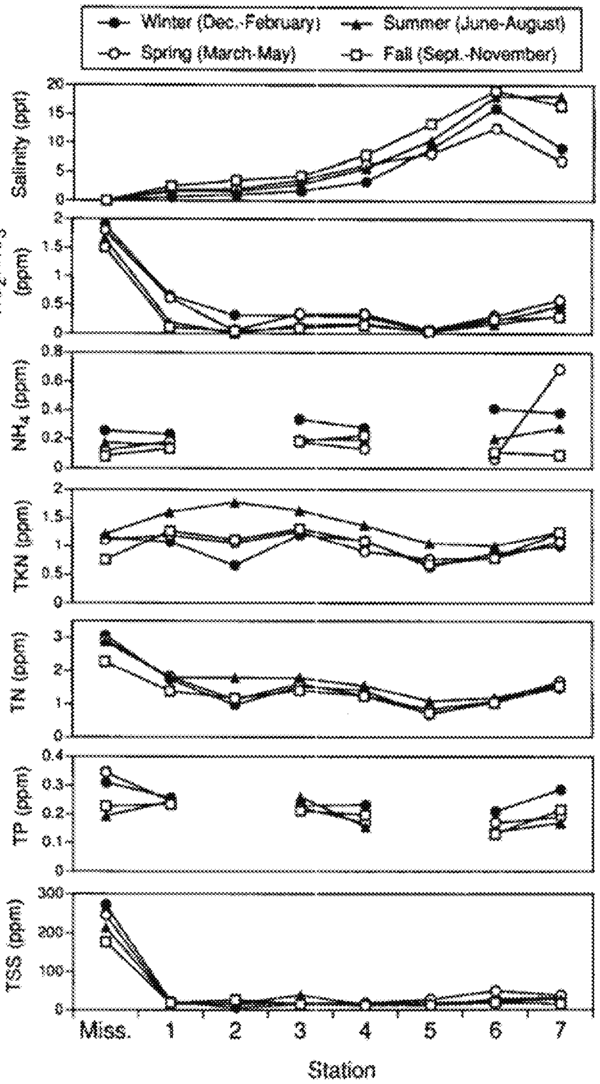

Fig. 5. Post diversion seasonal means, with standard error bars, for salinity, nitrite + nitrate, ammonium, total Kjeldahl nitrogen, total nitrogen, total phosphorous, and total suspended solids.

trations during summer and autumn were somewhat lower compared to winter and spring, but were not significantly different.

The BACI statistical analysis failed to find a significant impact on NH_4 concentrations due to the diversion (Fig. 1). Ammonium levels initially increased with distance from the diversion structure, then stabilized further into the estuary, and rose again presumably in response to freshwater water input at the Bohemia structure and spillway (Fig. 3). The salinity mixing diagram (Fig. 4) indicated the estuary was acting as a source for NH_4. Winter had the highest NH_4 concentrations compared to the other seasons (Fig. 5).

The BACI statistical analysis along with station by station contrasts indicated TKN levels at stations 2, 5, and 6 decreased due to the diversion (Table

TABLE 1. Results of pre/post-diversion by station statistical contrasts for variables Total Kjeldalh Nitrogen, Total Nitrogen, Total Phosphorus and Salinity. Level of significance: ****α = < 0.0001, ***α = 0.001, **α = < 0.01, *α = < 0.05, NS–no significant difference. The sign in the parenthesis indicates an increase (+) or decrease (−) as a result of the diversion.

Measured parameter	Station: River	1	2	3	4	5	6	7
TKN	NS	NS	**(−)	NS	NS	***(−)	**(−)	NS
TN	NS	NS	*(−)	NS	NS	***(−)	*(−)	NS
Salinity	NS	*(−)	****(−)	*(−)	**(−)	NS	NS	NS

TABLE 2. Water quality data of transects performed April 30 and June 21, 1996. The transects run from the Caernarvon diversion to Corps of Engineers water quality monitoring Stations 1 and 2.

	April 30, 1996		June 21, 1996	
Distance From Diversion (km)	$NO_2 + NO_3$ (ppm)	TSS (ppm)	$NO_2 + NO_3$ (ppm)	TSS (ppm)
0	0.54	277	2.14	115
4	0.44	177	1.35	15
Route To Station 1:				
7.5	0.37	36	0.29	5
10	0.41	41	0.10	7
18	0.16	27	0.17	12
Route To Station 2:				
7	0.57	109	1.48	43
14.5	0.50	18	0.08	12

1). Mean TKN concentrations were higher in the upper estuary (1.2–1.6 ppm) than in the Mississippi River (0.9–1.1 ppm), but decreased further into the system, suggesting TKN production in the upper estuary (Fig. 3). This was supported by the salinity mixing diagram (Fig. 4), which indicated the estuary was acting as source for TKN in the upper estuary. TKN is a combination of NH_4 and organic nitrogen (ON). Since NH_4 accounts for less than 15% of TKN concentration, it can be inferred the estuary was acting as a source for ON. The highest TKN concentrations occurred during summer, and the lowest concentrations during winter (Fig. 5); seasonal salinity mixing curves showed the estuary acting as a strong source for TKN during the summer.

The BACI statistical analysis along with station by station contrasts indicated TN levels at stations 2, 5, and 6 decreased due to the diversion (Table 1), the same as TKN. TN levels generally decreased through the marsh stations with elevated concentrations at station 7, presumably due to the influence of the Bohemia structure and spillway (Fig. 3). The salinity mixing curve was concave, indicating the estuary was acting as a sink for TN (Fig. 4), mostly due to $NO_2 + NO_3$ uptake. The summer had higher TN concentrations compared to other seasons, which had relatively uniform concentrations (Fig. 5), mostly due to TKN.

The BACI statistical analysis failed to find the diversion as having a significant impact on TP concentration (Fig. 3). Mean TP levels steadily decreased through the marsh stations with an increase at station 7, presumably due to the Bohemia structure and spillway (Fig. 3). The salinity mixing diagram indicated the system was generally conservative with respect to TP (Fig. 4). TP concentrations did not show distinguishable seasonal trends in the upper estuary (stations 1–3), but did display

TABLE 3. Nutrient loading rates and removal efficiency of wetlands north of the first two water quality monitoring stations.

	Year		
	1992	1993	1994
$NO_2 + NO_3$			
Loading (g m^{-2} yr^{-1})	5.6	7.3	13.4
% Removal	97	95	88
Total Nitrogen			
Loading (g m^{-2} yr^{-1})	8.9	12.1	23.4
% Removal	57	47	32
Total Phosphorus			
Loading (g m^{-2} yr^{-1})	0.9	1.4	2.0
% Removal	46	39	0

somewhat higher winter concentrations in the lower estuary (Fig. 5).

The BACI statistical analysis failed to find the diversion as having a significant impact on TSS concentration (Fig. 3). Mean TSS concentrations in the upper estuary were very low (< 20 ppm) compared to the Mississippi River (> 150 ppm), suggesting the estuary was rapidly assimilating introduced sediments (Fig. 3). This was confirmed by the very concave curvature of the salinity mixing diagram (Fig. 4), indicating the estuary was acting as a strong sink for TSS. In spring, the lower estuary (stations 6 and 7) had higher TSS concentrations, compared to the rest of the marsh probably due to wave resuspension (Fig. 5).

Results of the independent sampling carried out April 30 and June 21, 1996, indicated TSS levels rapidly decreased before reaching the first monitoring stations (Table 2). The same was true for $NO_2 + NO_3$ concentrations, except April 30 on the route to station 2 when $NO_2 + NO_3$ concentrations remained relatively unchanged. This data supports the hypothesis that the Caernarvon freshwater diversion has not affected TSS or $NO_2 + NO_3$ levels in Breton Sound because these constituents were either transformed, assimilated or deposited before reaching the first water quality monitoring stations.

Results of nutrient loading rate analysis indicated 5.6 to 13.4 g m^{-2} yr^{-1} of $NO_2 + NO_3$, 8.9 to 23.4 g m^{-2} yr^{-1} of TN, and 0.9 to 2.0 g m^{-2} yr^{-1} of TP were delivered into the region north of the first two water quality monitoring stations from the input of Mississippi River at Caernarvon during 1992 to 1994 (Table 3). Removal efficiencies for these constituents were 88 to 97% for $NO_2 + NO_3$, 32–57% for TN, and 0 to 46% for TP.

Discussion

The results of this analysis showed rapid reduction of $NO_2 + NO_3$ as river water entered the estuary. Various studies have reported similar reduction of $NO_2 + NO_3$ in estuarine environments with much of the reduction due to denitrification (Lindau and DeLaune 1991; Nowicki et al. 1997). Jenkins and Kemp (1984) reported that up to 50% of NO_2 + NO_3 introduced into the Patuxent River estuary underwent denitrification. This process is carried out by denitrifying bacteria that use nitrate as an electron acceptor to oxidize organic matter anaerobically (Koike and Hattori 1978). Another transformation pathway of NO_2 + NO_3 is assimilation into particulate organic matter by autotrophic photosynthetic organisms. Vascular plants as well as algae incorporate NO_2 + NO_3 into cellular mass. NO_3 reduction to NH_4 has also been found to occur (Smith et al 1982). Sorenson (1978) found as much as 50% of NO_3 applied to marine sediments can be reduced to NH_4. These processes are biologically driven and therefore are positively correlated to temperature, which may explain higher NO_2 + NO_3 concentrations during winter, when these processes were at their slowest rate.

The Breton Sound estuary acted as a source for NH_4. This was most likely caused by the regeneration of NH_4 by the decomposition of organic matter (Kemp and Boynton 1984), as well as reduction of NO_2 + NO_3 to NH_4 (Sorenson 1978). Bacteria and fungi decompose organic material to obtain energy and in the process release nutrients in dissolved organic form (Day et al. 1989). Numerous studies have shown the net mobilization of NH_4 by benthic sediments (Koike and Hattori 1978; Callender and Hammond 1982; Teague et al. 1988). The relatively shallow water depths, rapid settling rates and rapid bacterial utilization result in fairly short residence times for organic material in estuarine waters (Moran and Hodson 1989). Therefore, much of the regeneration of nutrients probably takes place on or in the sediments, which is where NH_4 regeneration is highest (Blackburn 1979).

Three of the seven water quality monitoring stations had decreased TKN and TN concentrations due to the diversion. One reason for this behavior could be simple dilution due to the addition of river water, which has lower concentrations of TKN than the receiving estuary. The estuary was acting as a source for TKN, especially during summer. This was most likely the result of higher primary production associated with increased nutrient input, along with an increased microbial community due to detritus availability, both of which are highest during the summer.

Overall, the Breton Sound wetlands acted as a sink for nitrogen. TN is a combination of all the forms of nitrogen discussed above, and its behavior is governed by processes controlling all those constituents. The high summer TN levels were probably due to increased primary and microbial productivity which also effected summer TKN concentrations. Denitrification plays a large role in the loss of nitrogen from estuarine waters, but another permanent loss of nitrogen is through burial of organic material such as detritus and senescent phytoplankton cells. Relative sea level rise in coastal Louisiana is approximately 1 cm yr^{-1} (Penland and Ramsey 1990), which is partially compensated for by an accretion rate of 0.7–0.9 cm yr^{-1} (Cahoon and Turner 1989; DeLaune et al. 1989). DeLaune et al. (1981) studied wetlands in Barataria Bay, which were accreting at a rate of 0.75 cm yr^{-1}, and found nitrogen was buried in the interior marsh at a rate of 13.4 g m^{-2} yr^{-1}. This burial rate is in the same range as the loading rate was for the region north of the first two water quality monitoring stations in Breton Sound estuary (8.9 to 23.4 g m^{-2} yr^{-1}), and where 32% to 57% nutrient removal efficiency was found.

The results indicate that Breton Sound estuary acted conservatively with respect to TP. The highly charged phosphate anion, PO_4^-, is readily sorbed by clay and detrital organic particles at high concentrations, while at lower concentrations PO_4^- is released into the water, thus maintaining moderate ambient concentrations (Jitts 1959). This buffering process occurs in two distinct steps, with sorption/desorption of PO_4^- directly onto particle surfaces occurring very rapidly (minutes-days), and then slow (days-months) diffusion of PO_4^- toward the interior of the particles (Froelich 1988). Cyclic aerobic and anaerobic conditions in the top several centimeters of the wetland soil also affect the sorption and release of phosphate (Patrick and Khalid 1974). Sharp et al. (1982) found these sorption-desorption processes provide a buffering mechanism for phosphorus in the Delaware estuary. Madden et al. (1988) showed that TP behaved similarly in Fourleague Bay, Louisiana, with little change in concentration throughout the year. The same processes most likely account for the conservative behavior of the Breton Sound estuary with TP. Burial is the only mechanism by which phosphorus is permanently lost from the system (Richardson 1985). Flocculation, and subsequent deposition, of dissolved organic and inorganic matter during the mixing of river and sea water is an important removal mechanism for phosphorus (Sholkovitz 1976). The TP loading rate for the region north of the first two water quality monitoring stations ranged from 0.9 to 2.0 g m^{-2} yr^{-1}, higher than the interior marsh burial rate found by DeLaune et al. (1981) of 0.8 g m^{-2} yr^{-1}, which may explain the low removal efficiency of 0% to 46% found in the Breton Sound estuary.

Suspended sediments introduced from the diversion were rapidly trapped in the Breton Sound estuary. This was due to decreasing water velocity when entering the estuary, allowing suspended sediment to drop out of the water column. Water passing through the Caernarvon diversion structure immediately enters Big Mar, an abandoned reclamation project, where sediment accumulation has been evident. Villarrubia (1998) reported 164 ha of new marsh has formed in Breton Sound estuary since 1991. Inorganic sediment is deposited in the marsh when the marsh is flooded. In coastal Louisiana, strong southerly winds associated with the prefrontal phase of winter storm fronts play a key role in raising water levels and distributing available sediments into the marsh interior (Reed 1989). These storm fronts resuspend bay bottom sediments and were probably responsible for increased TSS levels in the lower estuary, which has a large fetch from the southeast and is greatly influenced by southerly winds. Stern et al. (1991) found the highest sediment retention in marshes surrounding Fourleague Bay, Louisiana, was during the winter and early spring. The same pattern of seasonal deposition also occurred in Barataria Basin, Louisiana (Baumann et al. 1984; Madden et al. 1988). Though a discontinuous and variable process, storm fronts are an important factor in delivering sediments and maintaining marsh elevation in the face of relative sea level rise. Day et al. (1995) discussed similar pulsing events as a dominate force in the formation and maintenance of the Rhone delta, France.

The loading rate analysis found efficient nutrient removal at low loading rates, but removal efficiency decreased rapidly with increasing loading rate. Richardson and Nichols (1985) reviewed a number of wetland wastewater treatment systems and found similar results. The nutrient loading rates reported here were lower for corresponding removal efficiencies for total phosphorus and total nitrogen than those reported by Richardson and Nichols (1985). This difference may be due to chemical differences between wastewater and Mississippi River water, and physiological differences of the receiving wetlands. The trend of low removal efficiency at high loading rates may be caused by shorter residence times of water in the estuary during high loading rates, allowing less time for biogeochemical processes to occur.

Conclusions

TKN, TN, and salinity were the only variables measured for this study that were significantly impacted by the Caernarvon freshwater diversion project. The receiving estuary experienced decreases in salinity levels, which was the original goal of the diversion project (Chatry et al. 1983). Suspended sediments were rapidly trapped by the ecosystem within the first few kilometers of the diversion structure. Approximately 164 ha of new marsh has been created so far by the introduction of river water into the system (Villarrubia 1998). The rapid assimilation of nutrients without significant change in dissolved oxygen concentration, as well as the rapid removal of suspended sediments, suggests more water can be allowed to flow into to the estuary without detrimental effects to the ecosystem. The United States Corps of Engineers analysis of fecal coliform, heavy metals and pesticides found no significant impact during the 1991–1994 post-diversion monitoring phase (United States Corps of Engineers 1995). The decision to allow a higher flow rate depends on management goals. If the primary goal is to manage salinity regimes for oyster production, this goal is being met with the current discharge. But if the goal is to distribute nutrients and sediments into the lower reaches of the estuary, in an effort to compensate for RSLR and wetland loss, a much larger discharge is necessary along with the construction of additional diversions or enlargement of the current structure. Turner and Boyer (1997) found many smaller diversions to be more cost effective than a single large diversion structure, and Mossa (1996) suggested the management of diversion flow regimes to coincide with maximum suspended sediment load of the Mississippi River. The management of riverine input into the estuarine environment should include a comprehensive monitoring program to asses the effectiveness of this management strategy as the estuary responds to increased nutrient and sediment availability with changes in vegetation type, flow paths, and primary productivity. With current attention being focused on the most efficient means to deliver fluvial sediments to wetlands, our results show that it is possible to utilize freshwater diversions as a valuable management tool in the preservation coastal Louisiana.

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Literature Cited

Barry, J. M. 1997. Rising tide, the great Mississippi flood of 1927 and how it changed America. Simon and Schuster Inc., New York.

Baumann, R. H., J. W. Day, Jr., and C. A. Miller. 1984. Mississippi Deltaic wetland survival: Sedimentation versus coastal submergence. *Science* 224:1093–1094

Blackburn, T. H. 1979. Method for measuring rates of NH_4 turnover in anoxic marine sediments, using a ^{15}N-NH_4 dilution technique. *Applied and Environmental Microbiology* 37(4): 760–765.

Cahoon, D. R. and R. E. Turner. 1989. Accretion and canal impacts in a rapidly subsiding wetland II. Feldspar marker horizon technique. *Estuaries* 12(4):260–268.

CALLENDER, E. AND D. E. HAMMOND. 1982. Nutrient exchange across the sediment-water interface in the Potomac River Estuary. *Estuarine, Coastal and Shelf Science* 15:395–413.

CHATRY, M. AND D. CHEW. 1985. Freshwater diversion in coastal Louisiana: Recommendations for development of management criteria. *In* 4th Coastal Marsh and Estuary Management Symposium 71–84.

CHATRY, M., R. J. DUGAS, AND K. A. EASLEY. 1983. Optimum salinity regime for oyster production on Louisiana's state seed grounds. *Contributions in Marine Science* 26:81–94.

CLOERN, J. E. 1987. Turbidity as a control on phytoplankton biomass and productivity in estuaries. *Continental Shelf Research* 7:1367–1381.

DAVIS, D. W. 1993. Crevasses on the lower course of the Mississippi River. *Coastal Zone* 1:360–378.

DAY, J. W., C. A. S. HALL, W. M. KEMP, AND A. YANEZ-ARANCIBIA. 1989. Estuarine Ecology. Wiley-Interscience, New York.

DAY, J. W. AND P. H. TEMPLET. 1989. Consequences of sea Level rise: Implications from the Mississippi Delta. *Coastal Management* 17:241–257.

DAY, J. W., D. PONT, P. F. HENSEL, AND C. IBANEZ. 1995. Impacts of sea-level rise on deltas in the Gulf of Mexico and the Mediterranean: The importance of pulsing events to sustainability. *Estuaries* 18(4):636–647.

DELAUNE, R. D., R. H. BAUMANN, AND J. G. GOSSELINK. 1983. Relationships among vertical accretion, coastal submergence, and erosion in a Louisiana Gulf Coast marsh. *Journal of Sedimentary Petrology* 53(1):0147–0157.

DELAUNE, R. D., C. N. REDDY, AND W. H. PATRICK. 1981. Accumulation of plant nutrients and heavy metals through sedimentation processes and accretion in a Louisiana salt marsh. *Estuaries* 4(4):328–334.

DELAUNE, R. D., J. H. WHITCOMB, J. W. H. PATRICK, J. R. PURDUE, AND S. R. PEZESHKI. 1989. Accretion and canal impacts in a rapidly subsiding wetland. I. Cs and Pb techniques. *Estuaries* 12(4):247–259.

FROELICH, P. N. 1988. Kinetic control of dissolved phosphate in natural rivers and estuaries: A primer on the phosphate buffer mechanism. *Limnology and Oceanography* 4:649–668.

GORNITZ, V., S. LEBEDEFF AND J. HANSEN. 1982. Global sea level trend in the past century. *Science* 215:1611–1614.

GREENBERG, A. E., R. R. TRUSSELL, L. S. CLESCERI, M. A. H. FRANSON, EDS. 1985. Standard Methods for the examination of water and wastewater. American Public Health Association. Washington D.C.

HATTON, R. S., R. D. DELAUNE, AND J. W. H. PATRICK. 1983. Sedimentation, accretion, and subsidence in marshes of Baritaria Basin, Louisiana. *Limnology and Oceanography* 28(3):494–502.

HATTON, R. S., W. H. PATRICK, AND R. D. DELAUNE. 1982. Sedimentation, nutrient accumulation, and early diagenesis in Louisiana Barataria Basin Coastal Marshes. *In* Kennedy (ed.), Estuarine Comparisons. Academic Press, New York.

JENKINS, M. C. AND W. M. KEMP. 1984. The Coupling of nitrification and denitrification in two estuarine sediments. *Limnology and Oceanography* 29(3):609–619.

JITTS, H. R. 1959. The adsorption of phosphate by estuarine bottom deposits. *Australian Journal of Marine and Freshwater Research* 10:7–21.

KEMP, W. M. AND W. R. BOYNTON. 1984. Spatial and temporal coupling of nutrient inputs to estuarine primary production: The role of particulate transport and decomposition. *Bulletin of Marine Science* 35(3):522–535.

KOIKE, I. AND A. HATTORI. 1978. Denitrification and ammonia formation in anaerobic coastal sediments. *Applied and Environmental Microbiology* 35(2):278–282.

LANE, R. R. 1998. Water Quality Analysis of a Freshwater Diversion at Caernarvon, Louisiana. M.S. Thesis, Louisiana State University, Baton Rouge, Louisiana.

LINDAU, C. W. AND R. D. DELAUNE. 1991. Dinitrogen and nitrous oxide emission and entrapments in *Spartina alterniflora* saltmarsh soils following addition of N-15 labeled ammonium and nitrate. *Estuarine, Coastal and Shelf Science* 32: 161–172.

LISS, P. S. 1976. Conservative and non-conservative behaviour of dissolved constituents during estuarine mixing, p. 93–130. *In* Burton and Liss (eds.), Estuarine Chemistry. Academic Press, New York.

L'HOMER, A. 1992. Sea-level changes and impacts on the Rhone coastal lowlands, p. 136–152. *In* M. A. Tooley, and S. Jelgersma (eds.), Impacts of sea-level rise on European coastal lowlands. Blackwell Publishers, Oxford, United Kingdom.

MADDEN, C. J., J. W. DAY AND J. M. RANDALL. 1988. Freshwater and marine coupling in estuaries of the Mississippi River deltaic plain. *Limnology and Oceanography* 33:982–1004.

MILLIMAN, J. D. AND K. O. EMERY. 1968. Sea levels during the past 35,000 Years. *Science* 162:1121–1123.

MORAN, M. A. AND R. E. HODSON. 1989. Formation and bacterial utilization of dissolved organic carbon derived from detrital lignocellulose. *Limnology and Oceanography* 34(6): 1034–1047.

MOSSA, J. 1996. Sediment dynamics in the lowermost Mississippi River. *Engineering Geology* 45:457–479.

NOWICKI, B. L., J. R. KELLY, E. REQUINTINA, AND D. V. KEUREN. 1997. Nitrogen losses through sediment denitrification in Boston Harbor and Massachusetts Bay. *Estuaries* 20(3):626–639.

PATRICK, W. H. AND R. A. KHALID. 1974. Phosphate release and sorption by soils and sediments: Effect of aerobic and anaerobic conditions. *Science* 186:53–55.

PENLAND, S., R. BOYD, AND J. R. SUTER. 1988. Transgressive depositional systems of the Mississippi delta plain: A model for barrier shoreline and shelf sand development. *Journal of Sedimentary Petrology* 58(6):932–949.

PENLAND, S. AND K. E. RAMSEY. 1990. Relative sea-level rise in Louisiana and the Gulf of Mexico: 1908–1988. *Journal of Coastal Research* 6(2):323–342.

PENLAND, S., H. H. ROBERTS, S. J. WILLIAMS, A. H. SALLENGER, D. R. CAHOON, ET AL. 1990. Coastal Land Loss In Louisiana. *Transactions Gulf Coast Association of Geological Societies* XL:685–699.

RABALAIS, N. N., W. J. WISEMAN, AND R. E. TURNER. 1994. Comparison of continuous records of near-bottom dissolved oxygen from the hypoxia zone along the Louisiana coast. *Estuaries* 17(4):850–861.

REED, D. J. 1989. Patterns of sediment deposition in subsiding coastal salt marshes, Terrebonne Bay, Louisiana: The role of winter storms. *Estuaries* 12(4):222–227.

REDDY, K. R., R. D. DELAUNE, W. F. DEBUSK AND M. S. KOCH. 1993. Long-term nutrient accumulation rates in the Everglades. *Soil Science Society of America Journal* 57:1147–1155.

RICHARDSON, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science* 228:1424–1427.

RICHARDSON, C. J. AND D. S. NICHOLS. 1985. Ecological analysis of wastewater management criteria in wetland ecosystems, p. 351–391. *In* Paul J. Godfrey (ed.), Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold Company, New York.

ROBERTS, H. H. 1997. Dynamic changes of the holocene Mississippi river delta plain: The delta cycle. *Journal of Coastal Research* 13:605–627.

RUSSELL, R. J. 1936. Physiography of lower Mississippi River Delta, p. 3–199. *In* Lower Mississippi River Delta, Reports on the Geology of Plaquemines and St. Bernard parishes. Department of Conservation, Louisiana Geological Survey, Geological Bulletin no. 8. New Orleans, Louisiana.

SCRUTON, P. C. 1960. Delta building and the deltaic sequence.

Recent Sediments, NW Gulf Coast of Mexico American Association of Petroleum Geologists Symposium (1960):82–102.

SHARP, J. H., C. H. CULBERSON, AND T. M. CHURCH. 1982. The chemistry of the Delaware estuary. General considerations. *Limnology and Oceanography* 27(6):1015–1028.

SHOLKOVITZ, E. R. 1976. Flocculation of dissolved organic and inorganic matter during the mixing of river water and seawater. *Geochemica et Cosmochimica Acta* 40:831–845.

SMITH, C. J., R. D. DELAUNE AND J. W. H. PATRICK. 1982. Nitrate reduction in *Spartina Alterniflora* marsh soil. *Soil Science Society of America Journal* 46(4):748–750.

SORENSON, J. 1978. Capacity for denitrification and reduction of nitrate to ammonia in a coastal marine sediment. *Applied and Environmental Microbiology* 35(2):301–305.

STERN, M. K., J. W. DAY AND K. G. TEAGUE. 1991. Nutrient transport in a riverine-influenced, tidal freshwater bayou in Louisiana. *Estuaries* 14(4): 382–394.

STANLEY, D. J. 1988. Subsidence in the Northeastern Nile Delta: Rapid rates, possible causes, and consequences. *Science* 240: 497–500.

TEAGUE, K. G., C. J. MADDEN, AND J. W. DAY. 1988. Sediment-water oxygen and nutrient fluxes in a river-dominated estuary. *Estuaries* 11(1):1–9.

TURNER, R. E. AND M. E. BOYER. 1997. Mississippi River diversions, coastal wetland restoration/creation and an economy of scale. *Ecological Engineering* 8:117–128.

TURNER, R. E. AND N. N. RABALAIS. 1991. Changes in Mississippi River water quality this century. *BioScience* 41(3):140–147.

UNDERWOOD, A. J. 1994. On beyond BACI: Sampling designs that might reliably detect environmental disturbances. *Ecological Applications* 4(1):3–15.

UNITED STATES CORPS OF ENGINEERS. 1995. Caernarvon freshwater diversion structure, hydrologic, water and sediment quality monitoring program comprehensive report. United States Corps of Engineers, New Orleans, Louisiana.

VILLARRUBIA, C. R. 1998. Ecosystem response to a freshwater diversion: the Caernarvon experience. Abstract. Symposium on Recent research in Coastal Louisiana: Natural System Function and Response to Human Influences. February 3–5, Lafayette, Louisiana.

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